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# Bird response to prairie wetland loss and to adjacent land-cover change differs depending on conversion type and spatial extent

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# Bird response to prairie wetland loss and to adjacent land-cover change differs depending on conversion type and spatial extent

A thesis submitted in partial fulfillment of the requirement
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by

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# Acknowledgments

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**Abstract:** My study explores 1) whether 10 species of prairie wetland birds respond to land-use land-cover change (LULC) in the area adjacent to their breeding habitat and 2) whether they respond differently to wetland loss depending on the replacement land-cover type at two spatial extents. I used Breeding Bird Survey data and adapted National Land Cover Database change data to quantify changes in bird species abundance and LULC. Models of the relationship between these variables suggest three things: 1) bird response to change in specific land-cover types varies on the basis of of the amount of the land-cover present initially, 2) response to habitat loss or adjacent LULC change varies at different spatial extents, 3) wetland loss to developed land is the most important for explaining change in avian abundance. My results suggest that conservation strategies for prairie wetland birds should regulate wetland transition to urban land cover more heavily than other land cover typess and that management strategies should take into account the amount of habitat and adjacent land-covers initially present when regulations are enacted.

## Introduction and Background

When conceptualizing my honors thesis, I was interested in exploring two ecological questions: 1) does change in land-use-land-cover (LULC) adjacent to a species' primary habitat influence its use, and 2) do species' responses to reduction in primary habitat change depending on the type of land cover replacing the habitat? In my attempt to address these far-reaching concepts, I explored two more specific questions. First, do wetland birds in prairie regions of the United States respond to change in the LULC matrix around their wetland habitat? Second, do these species respond differently to different types of wetland loss?

I chose to explore trends in bird populations for multiple reasons. Birds make effective bioindicators for damaged or recovering ecosystems (Ortega-Álvarez and Lindig Cisneros, 2012). Furthermore, point counts quantifying bird species abundance are available going back several decades. Although point counts are subject to error due to variations in detection probability dependent on observer and other factors, they are widely used in studies similar to mine (Rittenhouse et al., 2010; Scholtz et al., 2017; Matthews et al., 2011).

I was interested in LULC change as a form of habitat disturbance for two reasons. First, it is the modus operandi of humans to alter the landscape to fit their needs. Humans are the direct cause of land cover changes such as increasing urban sprawl, expanding agriculture, and dwindling forest cover (Vitousek et al., 1997). Second, we know from existing conservation and ecological research that neighboring ecosystems and land covers can influence each other's ecological functions via spillover effects (Hansson et al. 2005; Horn et al., 2005). Because many species use multiple land cover types for different behaviors, such as feeding and breeding (Rodewald, 2015), I hypothesized that species would respond to disturbance in land cover adjacent to their primary habitat types.

Many studies investigating habitat disturbance take place over just a few years or are relatively small in extent. They often examine responses of only a few small populations to known sources of disturbance, rather than looking for trends spanning several years (Coppedge et. al, 2001; Engle et al. 1999; Thompson et al., 2015; Murkin et al., 1997; Shutler et al., 2000). My study is unique because I address my questions at large spatial and temporal extents. My data were drawn from years between 1988 and 2017, and my study area included a large portion of the conterminous United States. Studies at extents as large as mine are particularly uncommon for non-forest birds. I found only a few studies exploring the relationship between bird populations and land cover at a spatial scale larger than a few states, and none that focused on prairie wetland birds (Rittenhouse et al. 2010; Scholtz et. al 2017; Matthews et al. 2011).

I chose to focus on prairie wetlands for two reasons. First, the Great Plains has a history of extreme land cover change. The Great Plains region of the United States has undergone an extreme shift in land use over the past 200 years due to the westward expansion of large human populations and later due to industrialized agriculture (Drummond et al., 2012). What was once primarily grassland is now a mosaic of natural covers such as prairie grass and scrub, wetlands, and forests; and anthropogenic land uses such as agriculture, livestock pasture, and urban development (Drummond 2012). Recent models predict that anthropogenic land cover types will expand in the Great Plains leading up to the year 2100, displacing natural land cover types that serve as habitat for many birds. Second, wetlands perform important ecosystem services and functions, such as nutrient retention, flood mitigation, and carbon sequestration, and contribute to high biodiversity within their boundaries and in the surrounding LULC matrix (Euliss et al. 2005; Hansson et al. 2005).

Here I used a two-decadal change in land cover adjacent to wetlands and changes in bird abundance to evaluate how wetland species respond to changes in wetland area and adjacent land cover extent. I also investigated which LULC change related best to variation in change in wetland bird abundance.

## Methods

### *Study Area*

I included North American Breeding Bird Survey routes from 11 of The Nature Conservancy's terrestrial prairie ecoregions: Central Shortgrass Prairie, Central Mixed-Grass Prairie, Central Tallgrass Prairie, Crosstimbers and Southern Tallgrass Prairie, Dakota Mixed-Grass Prairie, Fescue Mixed-Grass Prairie, Northern Great Plains Steppe, Northern Tallgrass Prairie, Osage Plains/Flint Hills Prairie, Prairie-Forest Border, and Southern Shortgrass Prairie. I excluded the Gulf Coast Prairies and Marshes ecoregion, because wetlands in this ecoregion consist mainly of salt marshes, which are populated by a different set of wetland bird species. I quantified LULC at two spatial extents around each route, BBS sampling (sampling hereafter) and watershed, to analyze how prairie wetland birds in these ecoregions responded to LULC change at two different spatial extents. The BBS sampling extent includes the 400 m buffer around each route within which bird species are surveyed. The watershed extent includes all of the small local watersheds that drain into the wetland systems along the BBS routes. I considered using species' average home ranges as an additional larger extent for analysis, but was unable to due to a lack of complete data.

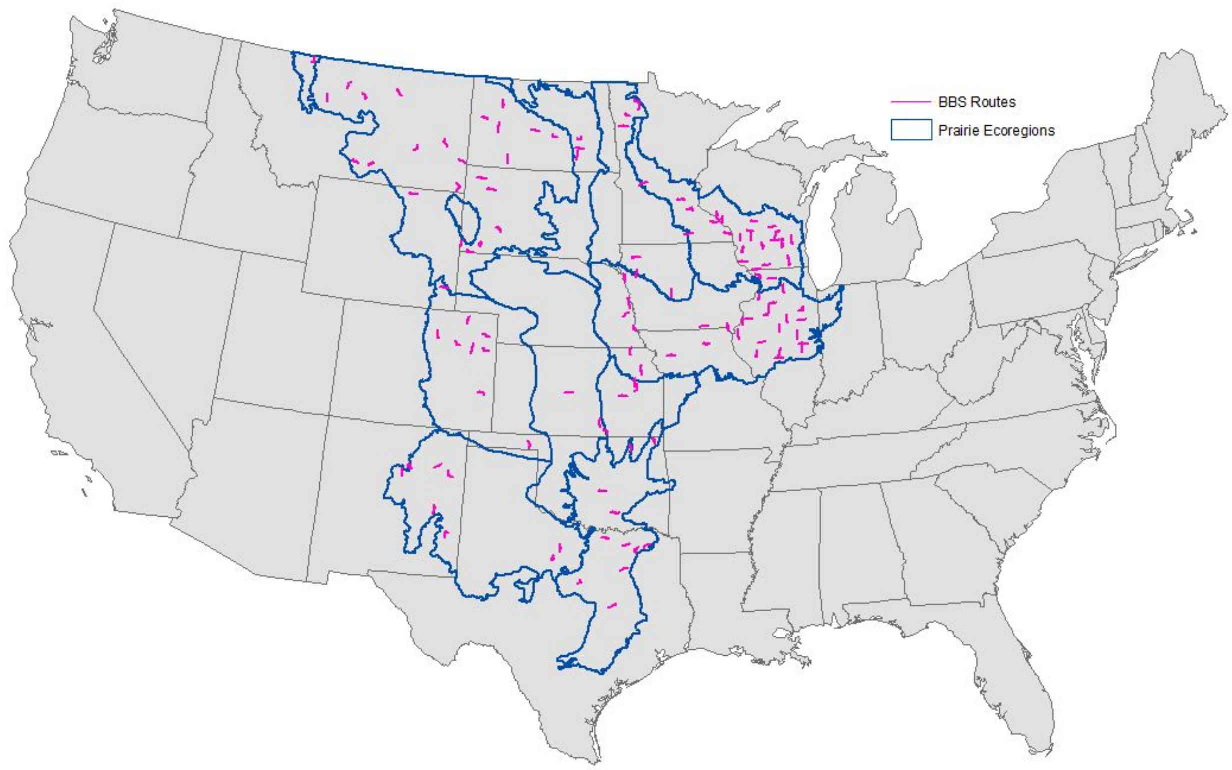


Figure 1: Study area defined 11 prairie ecoregions, and the 122 BBS routes included.

*Land-cover*

To compare land-cover change between 1992 and 2011, I used the NLCD 1992/2001 Retrofit Land Cover Change classification and the NLCD 2011 classification. Because of different sensors on satellites (Landsat 5 vs. Landsat 8) and different classification workflows, the original NLCD 1992 classification and the most recent 2011 classification are incomparable. To enable direct comparison (Price et al. 2003; Mogollón et al. 2016), I minimized error by adapting the 1992 Retrofit classification (Fry et al. 2009), which provides land-cover change information at the Anderson Level I classification scale (Anderson et al., 1976). I created a 1992 land cover layer in ENVI (Exelis Visual Information Solutions, Boulder, Colorado) by reclassifying the changed pixels of the NLCD Retrofit classification to their "from" value. I then reclassified the NLCD 2011 classification to match the eight 1992 classes , ascribing NLCD 2011 class codes to

Anderson Level I classifications in the same manner that the NLCD 2001 classification was ascribed in the creation of the Retrofit classification (Fry et al., 2009).

To create a land cover change layer from these two products, I followed the workflow shown in Figure 2. This resulted in a land cover change layer with persistent land cover and land cover change classes identical to the classes included in USGS Retrofit classification. The persistent classes were open water, developed, barren, forest, grass/shrubland, agriculture, wetland, snow/ice, and wetland, and the change classes represented all possible transitions between two land-cover types. Open water, barren land, forest, grass/shrubland, and snow/ice cover types were all fairly unambiguous classes. The developed class included roads, as well as urban development across a range from open space to high intensity; agriculture included cropland, hay, and pasture; and wetlands included woody wetlands and emergent herbaceous wetlands (Fry et al., 2009).

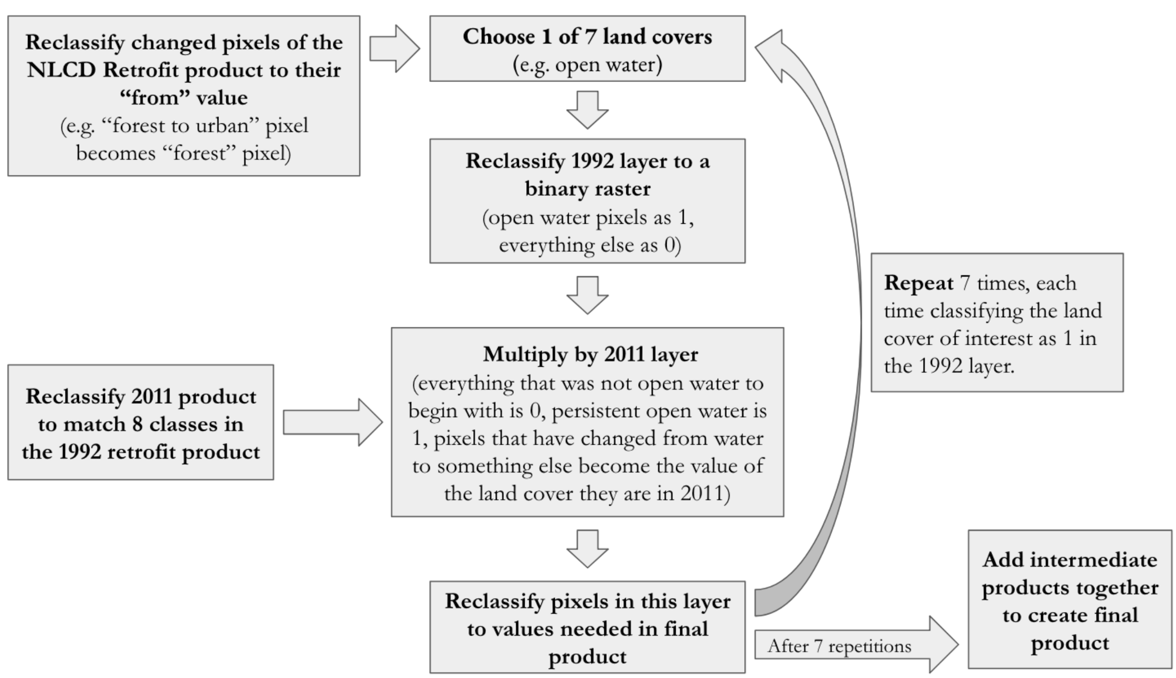

Figure 2: Workflow of creation of 1992 to 2011 land cover change layer.

To obtain land cover data within the two extents of my study, I used the following procedures. For the sampling extent, I buffered the final 122 BBS routes by 400 m, which is the maximum distance from the route at which species were detected (Sauer et al. 2017). I then merged these buffers based on the BBS route name to prevent buffers around intersecting routes from being split into multiple polygon features. For the watershed extent, I selected the USGS Hydrologic Unit Code 12 (HUC 12 henceforth) watersheds that fully enclosed the BBS routes. I merged those watersheds based on the BBS route name to create 122 polygons representing the watersheds that fed into the wetland systems along each BBS route. The borders of these local scale watersheds represent the largest area of neighboring land-use matrix that could be considered to affect the wetland-bird habitats. An example of the two extents and the land-cover within them can be seen in Figure 3.

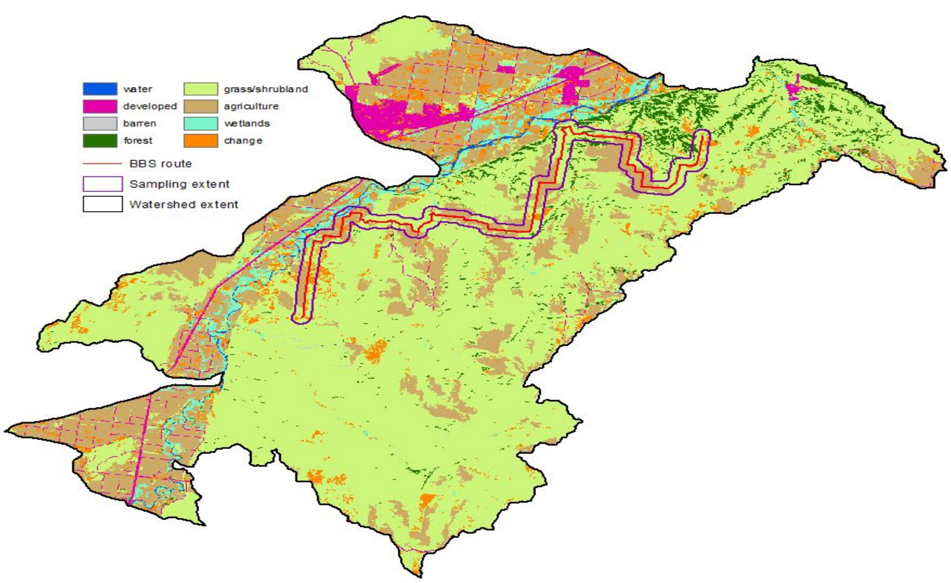


Figure 3: Example of the LULC data and the two extents for one BBS route included in the study.

All of the polygons surrounding each route differed in area because the BBS routes were not all of the same length and the watersheds were not of the same size. To adjust for difference in area sampled, I chose to calculate the proportion of each land cover type within each extent. I summed the total area of each land cover type to calculate the total area within each sampling and watershed extent, and used those data to calculate the proportion of each type of land cover and land cover change at each route at the two extents. I combined change classes and the proportion of persistent land cover for each type to calculate the proportion of each land cover type present in Period 1 and Period 2. I then used these values to calculate the overall change in proportion. A negative value signified overall loss of a land cover between periods, while a positive number signified gain. Although it is included in graphs of LULC, I excluded the barren land cover type from the models entirely because it is often transient, appearing and disappearing at a smaller temporal scale than I assessed in my study. Furthermore, barren cover accounted for little area within the extents of my study, and did not exhibit any pattern of growth or decline in the United States between 1973 and 2000 (Sleeter et. al, 2013). Ice/snow were not present within the extent of my study. All spatial analyses were conducted in ArcMap 10.4.

*Avian abundance*

I developed an initial list of prairie wetland bird species to include in my analysis based on a paper by Brown and Dinsmore's on marsh bird management in prairie wetlands (1986). I ultimately removed 15 species that were observed on fewer than 30 routes, due to sample size restrictions of my modeling process. This left 10 species: the Red-winged Blackbird (*Agelaius phoeniceus*), Common Grackle (*Quiscalus quiscula*), Mallard (*Anas platyrhynchos*), Common Yellowthroat (*Geothlypis trichas*), Canada Goose (*Branta canadensis*), Wood Duck (*Aix sponsa*), Blue-winged Teal (*Spatula discors*), Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), Sedge Wren (*Cistothorus platensis*), and Sora (*Porzana Carolina*).

I estimated changes in bird abundance between 1992 and 2011 using The North American Breeding Bird Survey (BBS), a cooperative, long-term avian survey effort, administered by the

USGS Patuxent Wildlife Research Center and the Canadian Wildlife service. Since 1966, data were collected along thousands of roadside survey routes by dedicated volunteers every year during avian breeding season. Each route is approximately 39.2 km long, with 50 stops spaced approximately 800 m apart along its length. During a survey, a volunteer conducts 3 minute point counts at each stop, recording every bird seen or heard within a 400 m radius. The routes are surveyed once per year during the breeding season (Pardieck, 2001). I created a candidate list of routes that included all 594 BBS routes completely contained within the 11 ecoregions I included in my study.

*Data cleaning*

I then downloaded for each of 564 routes counts of the total number of individuals and associated environmental data for 1987 through 2017 for all available species. The resulting data set contained 673,597 rows of data, with each row representing the observed counts of one species on a specific route in a specific year. I merged the environmental data associated with the count data.

I followed standard BBS data cleaning procedures (Rittenhouse et al., 2010; Pardieck, 2001) to remove all observations that were collected under suboptimal conditions. I removed all route-year observations where the wind speed at the beginning or end of the sample exceeded a 2 on the Beaufort scale, which brought the number of rows to 383,777. I then removed all observations where the “sky code” (a representation of weather) at the beginning or end of the route corresponded with drizzle, rain, or fog conditions. This brought the number of observations down to 362,083. Finally, I removed all route observations conducted by an observer in their first year, bringing the number of rows down to 304,694.

*Selection of time periods and years of data*

To compare land cover and bird abundance at snapshots of time, 1992 and 2011, using LULC in the matrix surrounding a given wetland habitat, I had to match temporal intervals of when satellite imagery and bird data were collected. The land cover data represented by the NLCD for 1992 and 2011 classification was derived from multiple Landsat satellite scenes that were merged to create full coverage. Scenes from 1987 through 1993 (period 1) were used in the creation of the 1992 Retrofit classification (Fry et al., 2009) and scenes from 2009 to 2011 (period 2) were used in the creation of the NLCD 2011 Product (U.S. Geological Survey, 2001 edition).

Because of the wide range of years represented within the land cover data, it would be inappropriate to only use BBS data from 1992 and 2011, despite the NLCD monikers attached to the layers. The BBS data in these years would not be a representation of the changes that could have occurred within the populations during periods 1 and 2. I found it necessary to account for that within-period change when attempting to analyze the change in populations that occurred between periods.

To decide which years of BBS data should be included in the model, I began with the range of years represented in the NLCD layers, 1987-1993 and 2009-2011. I then shifted each of these periods forward by one year to 1988-1994 and 2010-2012, to account for the fact that most scenes used in the land cover classification would have been from later in the year the BBS surveys are conducted (Pardieck, 2001; Fry et al., 2009), so the bird populations each year are likely experiencing a landscape more similar to the one represented a year later on the spatial data.

Several studies have asserted the existence of a time lag between when a habitat disturbance occurs and when bird populations respond (Schmiegelow and Mönkkönen, 2002;

Wiens et al., 1986; Chamberlain et al., 2000). Ideally, a study attempting to assess avian response to disturbance would incorporate knowledge of the philopatry, site fidelity, and survivorship of a species (Wiens et al., 1986). I had hoped to find relevant data for all of my species to incorporate into my analysis, but was again foiled by lack of data.

Due to the removal of routes during the data cleaning process and missing surveys due to lack of field volunteers, BBS data were not available for every route from every year. Furthermore, including every year within each period would be inappropriate; because period 1 includes seven years and period 2 includes only three, including every year would result in unequal sampling effort between periods. To resolve these problems, I extended period 2 to include data from 2010 to 2017, because this is the extent of the data available. I then extended period 1 to match in length, resulting in a period from 1988 to 1995. This increased the base of years from which I could pull bird data, which increased the number of routes I was ultimately able to include and helped me account for lag in bird response to habitat suitability. Once I had established these final periods, I pulled out only routes that had at least three years of sampling effort within each period, resulting in 122 routes. For routes where more than three years of data within a period were available, I randomly selected three years of the data in each period to include in the final dataset.

## Statistical Analysis

### *Change in bird abundance*

Not all of the species' breeding ranges covered the entire area included in my study, so I chose to analyze each species independently instead of using a richness metric. For each species, I averaged the total number of individuals observed on each route across the 3 years that fell within each of the 2 time periods, and then subtracted the means in the first period from those in the second. For a given species, negative values represented a decrease in mean abundance and positive values represented an increase.

I assessed the normality of differences in mean abundance on the basis of Q-Q plots. All species showed deviation from normality for extreme changes in abundance. Therefore, I used the Johnson transformation to approximate a normal distribution (Johnson, 1945; Slifker and Shapiro, 1980). I confirmed that the transformed data were more normally distributed using Q-Q plots and by examining plots of the residuals of models.

*Modeling framework*

To assess changes in species abundance over time, I fit mixed effects models in R Statistical Software (R Core Team, 2013) using the lme4 library (Bates et al., 2015). This model framework allowed me to combine fixed-effects terms for land cover change with a random effect term for the TNC ecoregion. This random effect helped to characterize variation in the response due to location within the study area and differences in ecological systems. At least 5 ecoregions were represented in the sample for every species, meeting the minimum requirements for number of levels represented by the random effect (Bolker et al., 2009). I did not use route as a random effect because differences in mean abundance were not replicated at the route level (Bolker et al., 2009).

I began the modeling process by ranking basic, single land-cover models for all species using the information-theoretic approach (Burnham and Anderson, 2002). I ranked models on the basis of AIC values using the AICcmodavg library in R (Mazerolle, 2019). During this process, I discovered that an interaction model including the initial area of the land cover and the total amount of change was consistently stronger than a model including only the base terms or higher-order terms, across all species. I proceeded to run single land-cover models with this basic interaction structure for the open water, developed, forest, grass/shrubland, agriculture, and wetland land covers at both the sampling and watershed extents to determine which extent better explained variation in the difference in mean abundance. I also evaluated interaction models for each type of wetland loss, including the initial amount of wetland, a specific wetland-to-other transition (e.g. wetland loss to developed land), and their interaction for all species. I selected the best extent for each land cover type or wetland transition on the basis of AIC values. If the AIC values for both extents were within 2 ΔAIC, I included the sampling extent model in the set of candidate variables.

I proceeded to more complex model structures with species that had a sample size of at least 60 routes, because I limited the number of fixed-effect parameters to no greater than 10% of the sample size to avoid overfitting. In these models, I always included the interaction between initial wetland cover and overall wetland change. I chose to include overall wetland change in all complex models because BBS routes are run during the nesting season of species, all of which nest primarily in or near wetlands (Rodewald, 2015). Depending on the model, I also included single land-cover change and wetland transition interaction models at their best extent. This resulted in models with the following basic structures:

Overall wetland and forest land cover change:

*Difference in mean abundance = Initial Wetland + Wetland Change + (Initial Wetland * Wetland Change) + Initial Forest + Forest Change + (Initial Forest * Forest Change)*

Overall wetland change and loss of wetland cover to forest cover:

*Difference in mean abundance = Initial Wetland + Wetland Change + Loss to Forest + (Initial Wetland * Wetland Change) + (Initial Wetland * Loss to Forest)*

Finally, for species that were observed on at least 90 routes, I included a third interaction. These models included the overall wetland change interaction, the overall water change interaction, and a third interaction term representing another type of land cover change or a type of wetland transition. I chose to include the overall water change interaction in all of these models because prior research on these species indicated that proximity to open water was important for all species (Rodewald, 2015). I wanted to explore whether other land covers or specific wetland transitions were important to explain variation in the difference in mean species abundance, even when the primary habitat of the species was accounted for. The basic structure of these models was as follows:

Overall land cover change (e.g. forest):

*Difference in mean abundance = Initial Wetland + Wetland Change + (Initial Wetland * Wetland Change) + Initial Water + Water Change + (Initial Water * Water Change) + Initial Forest + Forest Change + (Initial Forest * Forest Change)*

Wetland transition (e.g. loss of wetland to forest):

*Difference in mean abundance = Initial Wetland + Wetland Change + (Initial Wetland * Wetland Change) + Initial Water + Water Change + (Initial Water * Water Change) + Loss to Forest + (Initial Wetland * Loss to Forest)*

*Model Averaging*

After ranking the complex models for each species, I averaged the top-ranking models with AIC weights that summed to $\geq 0.95$. I followed standard model-averaging procedures by multiplying each parameter estimate and associated standard error by the model weight and summing the resulting values to obtain weighted averages of the estimates and standard error (Burnham and Anderson, 2002).

**Results**

After cleaning the BBS data, I was left with 122 routes. Four species were present on fewer than 60 routes: the Blue-winged Teal (n = 49), Yellow-headed Blackbird (n = 38), Sedge Wren (n = 36), and Sora (n = 30). Two species, Canada Geese (n = 81) and Wood Ducks (n = 81), were present on at least 60 routes but fewer than 90. Four species were present on at least 90 routes: the Red-winged Blackbird (n = 122), Common Grackle (n = 118), Mallard (n = 97), and Common Yellowthroat (n = 93).

*Land cover change*

At both extents, there was significantly more agriculture than any other cover type, followed by grass/shrubland (Figure 4). At the sampling extent, forest and developed land were the next most common and were present in comparable amounts. These were followed by wetlands, open water, and barren cover. At the watershed extent, the next most common land cover types were forest, wetland, and developed land. The confidence intervals for wetland overlapped with both, but there appeared to be significantly more forest than developed land. Open water and barren land were the least common land-cover types)

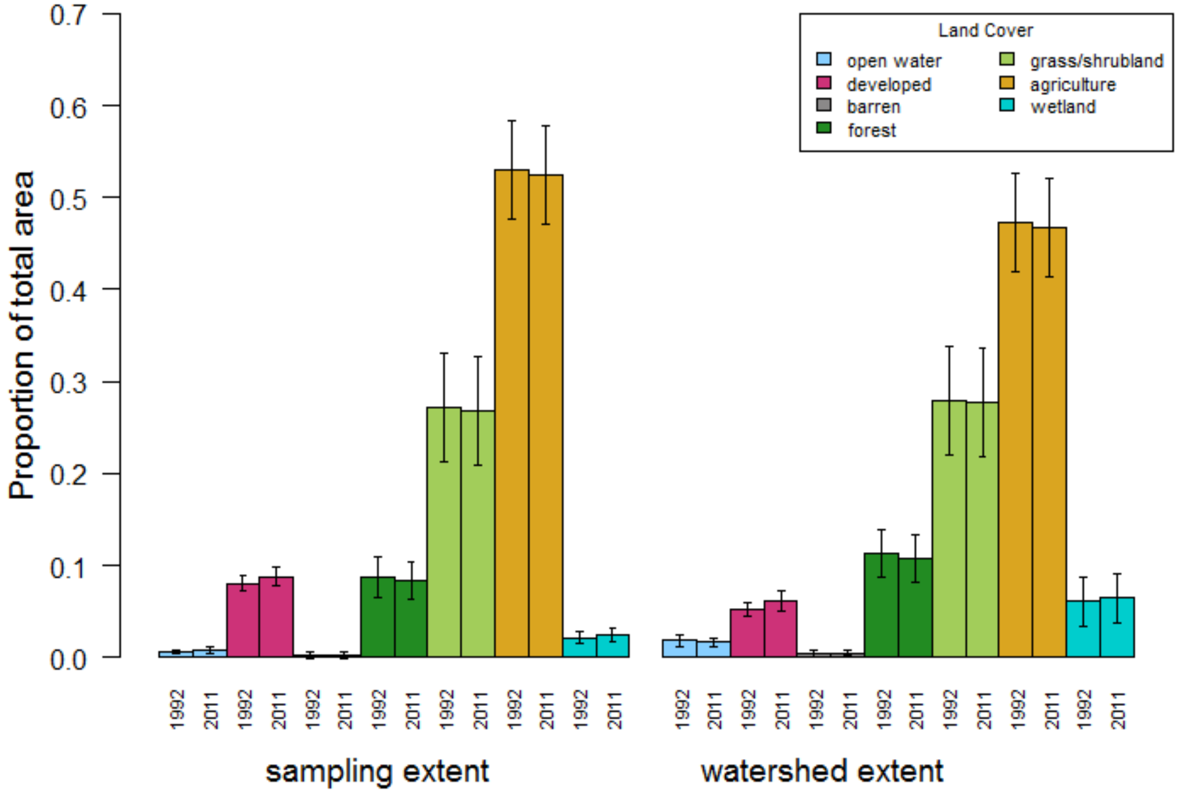


Figure 4: Comparison of the mean amount of each land cover type in 1992 and 20ll, at both extents. Error bars represent 95% confidence intervals for the mean.

The scale of Figure 4 is not ideal for discerning patterns in change, so Figure 5 demonstrates that the mean change in proportion of land cover differed greatly among types. On average, developed land, wetland, open water, and barren land significantly increased in proportion of the total area, but agriculture, grass/shrubland, and forest decreased. The only land covers for which change was not significant for both extents were grass/shrubland and water, for which the confidence intervals for mean crossed 0 at the watershed extent. Developed land showed the greatest change in any individual land cover.

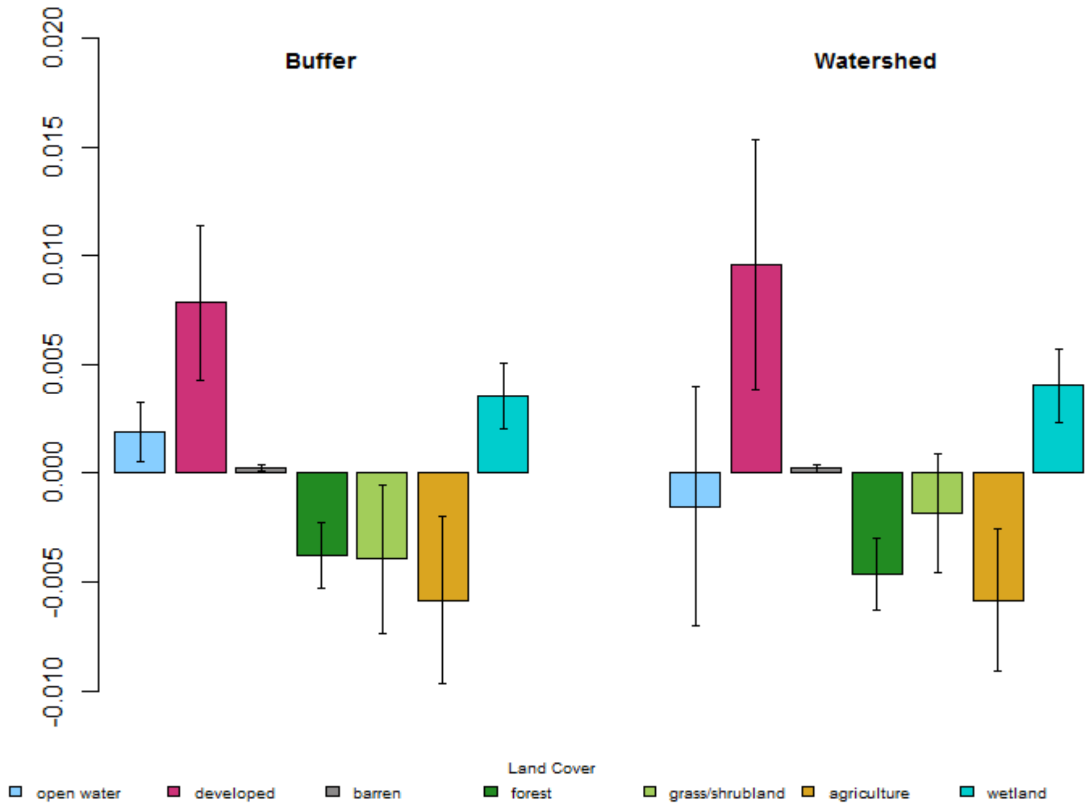


Figure 5: Proportion of total area attributed to mean change in land cover type across all routes, at both extents. Error bars represent 95% confidence intervals for the mean.

Although overall wetland area increased on average in both the sampling and watershed extents, this represents an exchange between areas of gain and loss. Figures 6 and 7 show the average percentage of wetland transition to six land cover types, across all routes for each of 10 species. Agricultural expansion and increase in open water accounted for the majority of wetland loss across all species, with grass/shrub land contributing a large but more variable amount to wetland loss. Significantly more wetland was lost to agriculture than to developed, forested, or barren land for all species at both extents. At the watershed extent, open water also accounted for significantly more loss than those three land covers for all 10 species, and significantly more wetland was lost to grass/shrubland for seven species. At the sampling extent, there was less significant difference among land cover types other than agriculture.

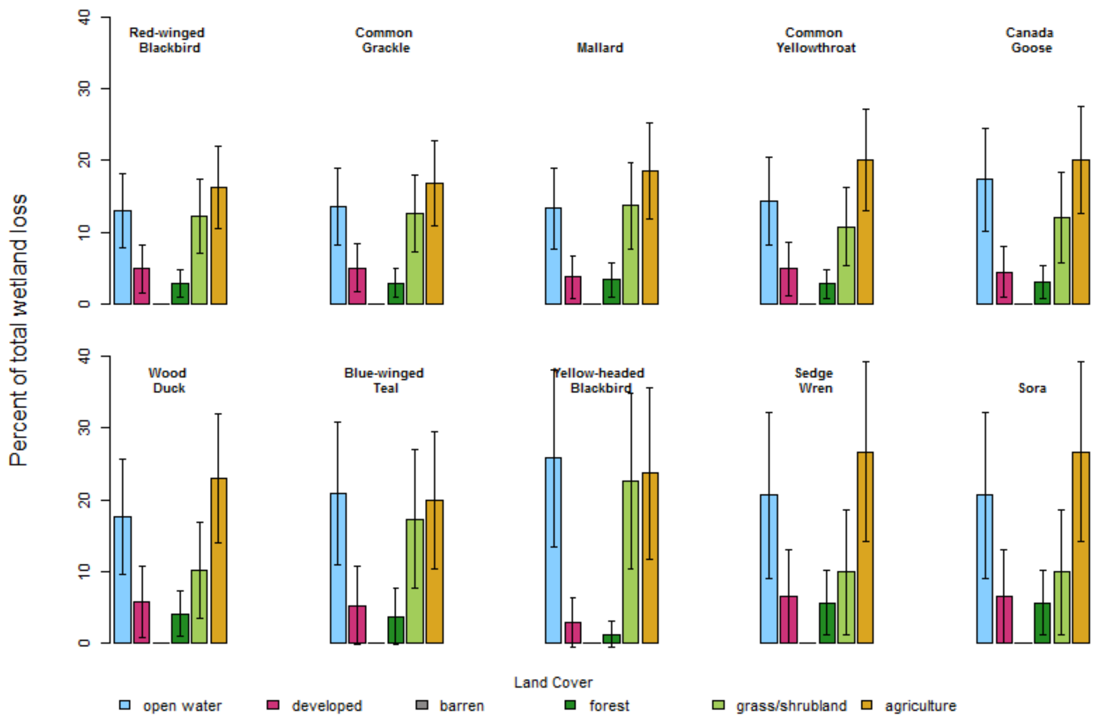


Figure 6: Percentage of total loss in wetland attributed to each land cover type on average across all routes inhabited by a species, at the sampling extent. Error bars represent 95% confidence intervals for the mean percentage.

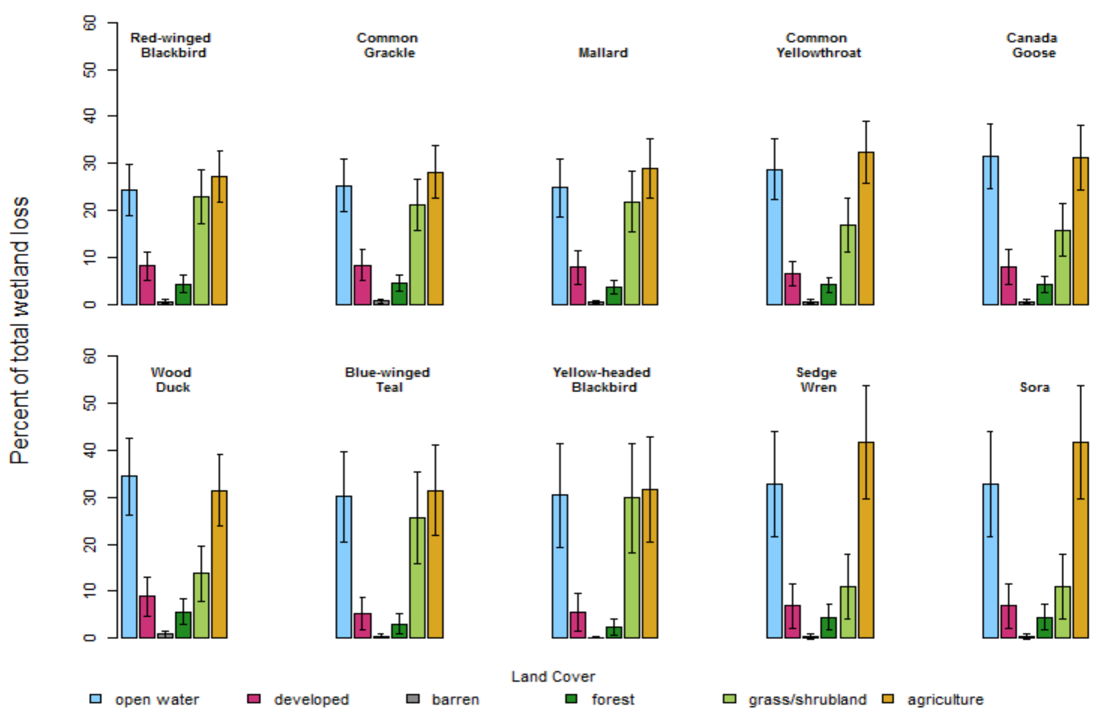


Figure 7: Percentage of total loss in wetland attributed to each land cover type on average across all routes inhabited by a species, at the watershed extent. Error bars represent 95% confidence intervals for the mean percentage.

*Models*

When relating change in mean species abundance to LULC, I found that all univariate, quadratic, and interaction models for all land cover types at both extents performed better than the null model. Because the interaction models were consistently stronger than univariate or quadratic models, I used these models to determine the best extent for a given land cover type and species. The sampling extent was more important than the watershed extent in 92% of models. The full suite of single land-cover models and wetland loss interaction models for each species can be found in Appendix 1.

For rare species, including the Sora, Sedge Wren, Yellow-headed Blackbird, and Blue-winged Teal, I was only able to assess single interaction models because these species appeared on fewer than 60 routes. The strongest single-cover and wetland transition interaction models for each species are shown in tables 1, 2, 3, and 4.

For the Sora, five models explain the variation in the difference in mean abundance (Table 1). Three of these models had similar, high AIC weights, and contributed most to explaining the variation in response. These were the models for loss of wetland to development, loss of wetland to grass/shrubland, and overall change in area of water. The interaction terms for all three of these models had a negative impact on the change in mean abundance. The slope of the interaction term for loss of wetland to developed land was steeper for a large area of wetland in 1992 compared to intermediate and small areas (Figure 8). The standard error of the estimate for the interaction term for wetland loss to developed land was greater than the estimate itself, suggesting imprecision in this estimate. One of the important models with a lower weight, wetland loss to water, had a positive estimate for the slope of the interaction term. For this interaction, the slope for a loss of wetland to water was positive for large initial amounts of wetland and negative for small and intermediate amounts (Figure 8).

Table 1: Sora, strongest single-cover and wetland transition models according to AIC weight. Overall change in amount of open water, as well as loss of wetland to developed land, grass/shrubland, water, and forest, all partially explain the change in abundance for the Sora. All models reported at sampling extent. The standard deviation of the random effect for eco-region ranges from 0.00 to 0.63.

| AICc | ΔAICc | AICcWt | Model Format: estimate(std. error) * parameter |
|---|---|---|---|
| 64.44 | 0 | 0.33 | -0.09(.4) + 2.41(5.79) * Initial Wetland - 304.9(1192) * Loss to Developed - 1261.0(10460.0) * (Initial Wetland * Loss to Developed) |
| 64.74 | 0.3 | 0.28 | -0.08(0.39) + 6.4(5.16) * Initial Wetland + 943.5(607.7) * Loss to Grass - 16780.0(9993.0) * (Initial Wetland * Loss to Grass) |
| 64.84 | 0.4 | 0.27 | -0.01(0.02) - 2.65(10.33) * Initial Water + 101.4(33.72) * Water Change - 6247.0(2465.0) * (Initial Water * Water Change) |
| 67.84 | 3.4 | 0.06 | 0.38(0.38) - 2.83(5.44) * Initial Wetland - 328.07(199.49) * Loss to Water + 2220.4(2509.55) * (Initial Wetland * Loss to Water) |
| 69.57 | 5.13 | 0.03 | 0.13(0.43) - 1.29(5.75) * Initial Wetland + 523.84(951.64) * Loss to Forest - 1911.72(3940.56) * (Initial Wetland * Loss to Forest) |

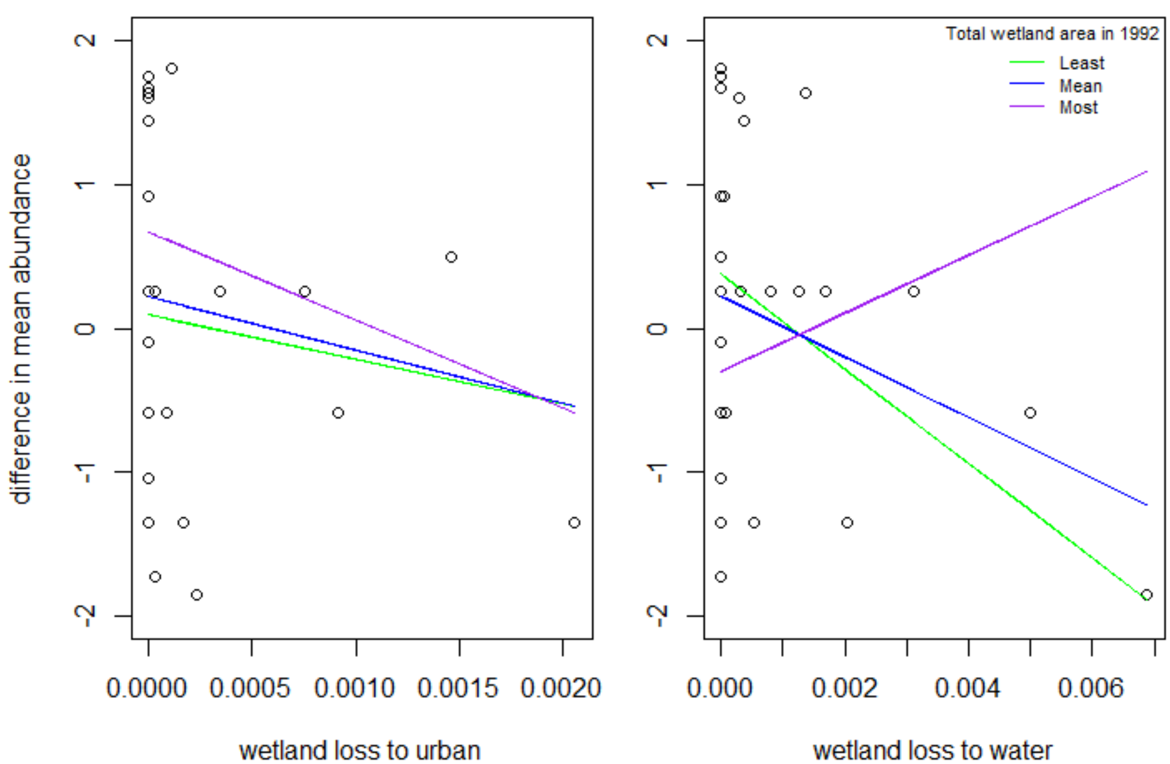


Figure 8: Graphs displaying the change in the effect of two types of wetland loss on the difference in mean abundance when the amount of wetland area in 1992 is held fixed at 3 values: the least amount of wetland present on any individual route, the mean amount present, and the largest amount present on any individual route. The interaction term for wetland loss to urban is negative for this species, and the interaction term for wetland loss to water is positive.

For the Sedge Wren, five models effectively explained the variation in the difference in mean abundance. The interaction models for loss of wetland to grass/shrubland, water, and developed land had the highest weights. Once again, all three interaction terms had negative estimates, and wetland loss to developed land was the only model for which the standard error of the parameter estimate for the interaction term exceeded the estimate itself.

Table 2: Sedge Wren: strongest single-cover and wetland transition models according to AIC weight. Overall change in amount of open water, as well as loss of wetland to grass/shrubland, water, developed land, and forest all partially explain the change in abundance for the Sedge Wren. All models reported at sampling extent. The standard deviation of the random effect for eco-region ranges from 0.5 to 0.69.

| AICc | ΔAICc | AICcWt | Model Format: estimate(std. error) * parameter |
|---|---|---|---|
| 67.28 | 0 | 0.43 | 0.03(0.34) + 4.99(5.13) * Initial Wetland + 950.57(690.06) * Loss to Grass - 9591.717(10514.92) * (Initial Wetland * Loss to Grass) |
| 68.7 | 1.42 | 0.21 | 0.15(0.35) + 6.23(3.89) * Initial Wetland + 479.6(412.8) * Loss to Water - 12860.0(8841.0) * (Initial Wetland * Loss to Water) |
| 69.28 | 2 | 0.16 | 0.2(0.38) + 4.24(4.27) * Initial Wetland + 414.0(865.21) * Loss to Developed - 4074.42(8755.71) * (Initial Wetland * Loss to Developed) |
| 69.99 | 2.72 | 0.11 | 0.25(0.3819) + 2.44(4.43) * Initial Wetland - 72.84(793(.38) * Loss to Forest + 2185.04(8211.4) * (Initial Wetland * Loss to Forest) |
| 71.4 | 4.12 | 0.06 | 0.5(0.38) - 19.65(30.15) * Initial Water - 32.04(17.6) * Water Change + 254.5(619.55) * (Initial Water * Water Change) |

Only 3 models effectively explained part of the variation in change in mean abundance for the Yellow-headed Blackbird. Loss of wetland to water and forest were once again included among these models, along with loss to forest. All interaction terms were negatively related to the response, and the coefficient of the wetland loss to developed land interaction was again exceeded by its standard error.

Table 3: Yellow-headed Blackbird: strongest single-cover and wetland transition models according to AIC weight. Loss of wetland to water, forest, and developed land all partially explain the change in abundance for the Yellow-headed Blackbird. All models reported at sampling extent. The standard deviation of the random effect for eco-region ranges from 0.58 to 0.86.

| AICc | ΔAICc | AICcWt | Model Format: estimate(std. error) * parameter |
|---|---|---|---|
| 75.06 | 0 | 0.69 | 0.39(0.35) + 0.08(3.39) * Initial Wetland - 158.7(448.3) * Loss to Water - 9358.0(8873.0) * (Initial Wetland * Loss to Water) |
| 77.42 | 2.36 | 0.21 | 0.27(0.43) - 4.77(6.21) * Initial Wetland - 1167.0(1340.0) * Loss to Forest - 12520.0(19080.0) * (Initial Wetland * Loss to Forest) |
| 79.55 | 4.49 | 0.07 | 0.28(0.41) - 3.5(6.12) * Initial Wetland - 478.26(1190.74) * Loss to Developed - 1368.29(10745.33) * (Initial Wetland * Loss to Developed) |

The variation in the change in mean abundance of the final rare species, the Blue-winged Teal, was partially explained by 5 relatively strong models. By far the strongest model was loss of wetland to developed land. The estimate of the parameter was negative and exceeded by its standard error. Loss to grass/shrubland and loss to forest were included among these strong models, as well as overall change in wetland. The parameters of these interactions were all negatively related to the response variable. The only model with an interaction that had a positive impact on change in mean abundance was the loss of wetland to water.

Table 4: Blue-winged Teal: strongest single-cover and wetland transition models according to AIC weight. Overall change in wetland and loss of wetland to developed land, water, grass/shrubland, and forest all partially explain the change in abundance for the Blue-winged Teal. All models reported at sampling extent. The standard deviation of the random effect for eco-region ranges from 0.48 to 0.65.

| AICc | ΔAICc | AICcWt | Model Format: estimate(std. error) * parameter |
|---|---|---|---|
| 90.24 | 0 | 0.54 | -0.11(0.28) + 0.74(3.69) * Initial Wetland + 340(728.68) * Loss to Developed - 3337.99(6454.69) * (Initial Wetland * Loss to Developed) |
| 92.22 | 1.98 | 0.2 | 0.05(0.25) + 0.36(3.83) * Initial Wetland - 177.12(122.36) * Loss to Water + 175.67(1787.38) * (Initial Wetland * Loss to Water) |
| 94.16 | 3.92 | 0.08 | -0.1(0.27) + 0.51(3.0) * Initial Wetland - 112.0(139.0) * Loss to Grass - 2390.0(2770.0) * (Initial Wetland * Loss to Grass) |
| 94.44 | 4.2 | 0.07 | -0.07(0.23) + 6.68(7.52) * Initial Wetland - 30.24(14.16) * Wetland Change - 879.48(321.91) * (Initial Wetland * Wetland Change) |
| 96.19 | 5.94 | 0.03 | -0.04(0.29) - 1.57(4.08) * Initial Wetland - 36.19(352.32) * Loss to Forest - 20.02(1432.41) * (Initial Wetland * Loss to Forest) |

For the remaining 6 species, I proceeded to more complex models. Tables of ranked complex models by species can be found in Appendix 1. Although the wetland-to-developed transition model appears most frequently as the strongest single-interaction model, the interactions I included in all complex models were aimed towards including primary habitat requirements for species, as discussed in the methods. The Wood Duck and Canada Goose were only present on enough routes to allow for 2 interaction terms, so for these species I was only able to combine the interaction model for overall change in wetland with one other single land-cover or wetland transition model in each complex model.

For the Wood Duck, 4 of these models contributed to explaining the variation in the mean change in abundance (Appendix 1, Table 11). The average of these models, below, indicates that the interactions between initial wetland and wetland loss to developed land, grass/shrubland, and water were most important in explaining the variation in the response. Although mean change in abundance decreased in response to an increase in loss of wetland to grassland, it actually increased in response to wetland loss to water and developed land. The standard deviation of the random effect for ecoregion ranged from 0.00 to 0.24 for the models included in the average. For this model and for all others reported, terms follow the format Estimate (Std. Error) * Parameter. Subscripts indicate the extent at which a parameter was assessed: B corresponds to the BBS sampling extent and HUC to the watershed extent.

$$\textit{Change in Mean Abundance} = -0.08(0.17) - 1.85(5.64) * \textit{Initial Wetland}_B + 26.8(30.94) * \textit{Wetland Change}_B - 560.45(591.8) * \textit{Loss to Developed}_B - 229.73(148.0) * \textit{Loss to Grassland}_B - 159.08(21.11) * \textit{Loss to Water}_B + 64.08(34.92)\ \textit{Loss to ForestB} - 180.58(442.62) * (\textit{Initial Wetland}_B * \textit{Wetland Change}_B) + 5676.00(7293.00) * (\textit{Initial Wetland}_B * \textit{Loss to Developed}_B) - 3{,}790.00(2{,}630.0) * (\textit{Initial Wetland}_B * \textit{Loss to Grassland}_B) + 3{,}936.15(351.06) * (\textit{Initial Wetland}_B * \textit{Loss to Water}_B) - 197.82(156.15) * (\textit{Initial Wetland}_B * \textit{Loss to Forest}_B)$$

For the Canada Goose, only 1 model (Table 12, Appendix 1) effectively explained the variation in mean change in abundance. This species responds most strongly to loss of wetland cover to developed land. It is interesting to note that more of the variation in the species response is explained by this change when it is evaluated at the extent of the watershed. Only on other species, the Common Yellowthroat, had a heavily weighted model that included a land cover change at this extent. The standard deviation of the random effect for ecoregion was 0.04.

$$Change\ in\ Mean\ Abundance = 0.11(0.14) - 3.94(4.79) * Initial\ Wetland_B + 41.75(21.65) * Wetland\ Change_B - 0.99(0.77) * Initial\ Wetland_{HUC} - 1396.26(384.46) * Loss\ to\ Developed_{HUC} + 39.57(241.97) * (Initial\ Wetland_B * Wetland\ Change_B) + 6568.67(2191.91) * (Initial\ Wetland_{HUC} * Loss\ to\ Developed_{HUC})$$

The Common Yellowthroat, Mallard, Common Grackle, and Red-winged Blackbird were all present on at least 90 routes, allowing the inclusion of a 3rd interaction in their models. All of these 3-interaction models included the single-interaction model for overall change in wetland and the single-interaction model for change in water, as well as one single land-cover or wetland transition interaction model.

AIC weights indicated that three of these models with 3 interaction terms partially explained the variation in mean difference in abundance for the Common Yellowthroat (Table 13, Appendix1). The average of these models indicated that overall change in water and wetland, as well as the loss of wetland to water and developed land, all contributed to explaining the variation in difference in mean abundance. Overall change in water and overall change in water cover both had a positive relationship with mean change in abundance. Single-cover models for this species (Table 7, Appendix 1) demonstrate that water cover explained Common Yellowthroat variation in response more strongly than wetland, so it is likely that this species was more reliant on water cover than wetland. Overall change in wetland and loss of wetland to developed cover both related negatively with difference in mean abundance. Unfortunately, the standard errors for all of these variables were greater in magnitude than the estimates, so it is possible that these variables did not truly help explain the variation in the species' response. The standard deviation of the random effect ranged from 0.00 to 0.31.

*Change in mean abundance* = $-0.07(0.15) + 1.43(4.61) * Initial\ Wetland_B + 30.01(28.7) * Wetland\ Change_B - 12.83(10.069) * Initial\ Water_B + 33.48(20.21) * Water\ Change_B + 0.55(0.14) * Initial\ Forest_{HUC} - 1.02(2.52) * Forest\ Change_{HUC} + 334.35(693.69) * Loss\ to\ Developed_B - 64.71(35.64) * Loss\ to\ Water_B + 267.38(395.72) * (Initial\ Water_B * Water\ Change_B) - 400.28(371.72) * (Initial\ Wetland_B * WetlandChange_B) - 440.44(8130.16) * (Initial\ Wetland_B * Loss\ to\ Developed_B) + 226.1(489.06) * (Initial\ Wetland_B * Loss\ to\ Water_B) + 25.64(11.51) * (Initial\ Forest_{HUC} * Forest\ Change_{HUC})$

Three models helped explain the variation in mean difference in abundance for the Mallard (Table 14, Appendix 1). The averaged model indicates that the interaction terms for loss of wetland to developed land and grass/shrub land contributed more heavily to explaining the variation in Mallard response than the interaction terms for overall change in water and wetland. While wetland loss to development was negatively related to difference in response, loss to grassland demonstrated a positive relationship with mean difference in abundance. Although the estimate associated with loss of wetland to developed land was the greatest in magnitude, it was also the only estimate exceeded by its associated standard error. The standard deviation of random effect for ecoregion ranged from 0.28 to 0.35 for models included in this average.

*Change in mean abundance* = $-0.11(0.18) + 1.03(4.66) * Initial\ Wetland_B + 79.6(26.44)\ Wetland\ Change_B - 8.35(9.56) * Initial\ Water_B + 19.27(20.06) * Water\ Change_B + 379.092112(700.38) * Loss\ to\ Developed_B - 45.4682608(27.49) * Loss\ to\ Grassland_B - 26.8(24.23) * Loss\ to\ Water_B - 404.5968072(355.71) * (Initial\ Wetland_B * Wetland\ Change_B) - 613.4989522(389.66) * (Initial\ Water_B * Water\ Change_B) - 5671.2(8215.83) * (Initial\ Wetland_B * Loss\ to\ Developed_B) + 795.829216(561.82) * (Initial\ Wetland_B * Loss\ to\ Grassland_B) - 3.62(323.01) * (Initial\ Wetland_B * Loss\ to\ Water_B)$

For the Common Grackle, only the model including the interaction between initial wetland and loss of wetland to open water helped explain the variation in mean change in abundance. The extreme slope and relatively low standard error associated with this parameter indicated a strong, positive relationship between wetland loss to water and change in mean abundance. The standard deviation of the random effect for ecoregion was 0.52.

$$\textit{Change in mean abundance} = 0.02(0.21) - 1.34(4.53) * \textit{Initial Wetland}_B + 36.22(23.32) * \textit{Wetland Change}_B + 6.97(8.79) * \textit{Initial Water}_B + 36.07(8.79) * \textit{Water Change}_B - 302.28(161.2) * \textit{Loss to Water}_B - 227.14(256.58) * (\textit{Initial Wetland}_B * \textit{Wetland Change}_B) - 764.78(365.72) * (\textit{Initial Water}_B * \textit{Water Change}_B) + 3253.33(161.2) * (\textit{Initial Wetland}_B * \textit{Loss to Water}_B)$$

The best complex model for the Red-winged Blackbird had an AIC weight of 0.94. Because this weight was very close to the cutoff of 0.95, I did not average this model with any others for this species. By far the steepest slope for a parameter was associated with the wetland loss to developed land interaction term. The change in mean abundance was negatively related to this parameter. The standard deviation of the random effect was 0.35.

$$\textit{Change in mean abundance} = -0.06(0.02) + 1.25(4.3) * \textit{Initial Wetland}_B + 27.04(26.7) * \textit{Wetland Change}_B - 10.61(9.29) * \textit{Initial Water}_B + 19.08(18.71) * \textit{Water Change}_B - 1581.0(958.6) * \textit{Loss to Developed}_B + 339.7(384.4) * (\textit{Initial Wetland}_B * \textit{Wetland Change}_B) - 759.2(375.8) * (\textit{Initial Water}_B * \textit{Water Change}_B) - 26450.0(11340.0) * (\textit{Initial Wetland}_B * \textit{Loss to Developed}_B)$$

**Discussion**

Very few studies have investigated spill-over influences of land-cover change adjacent to wetland species' primary habitat (T Findley and Bourdages, 2000; Petersen and Westmark, 2013). For wetland species, I found that LULC change and wetland loss at the sampling extent were more important in explaining the variation in the difference in mean abundance than at the watershed extent. Second, I found that the interaction between the initial amount of land cover and the amount it changed was more important than either factor on its own. Third, I found that loss of wetland to developed land was the most important type of wetland loss, followed by loss to grassland and forest.

Variation in change in mean abundance related to LULC change at the sampling extent for 8 of 10 species. Only two birds in my study, the Common Yellowthroat and the Canada Goose, responded to land cover change at the watershed extent. Therefore, I infer that a species' primary habitat and changes adjacent to that area more heavily influence change in abundance than change at a watershed extent. However, it is important to note that all of the models of LULC change at the watershed extent were better than the null model. Although these models were outweighed by models including land-cover at the sampling extent, change at that scale did have some effect on species' abundance. This variation in response to habitat variables at different spatial extents has been also been noted in studies on other wetland species and in bird studies with smaller study areas (Brown and Dinsmore, 1986; Price et al., 2004; Rooney et al., 2012).

The effect sizes and directionality of slopes of the interaction terms in my models suggest that species' responses to wetland loss vary based on how much wetland was present previously. The initial area of wetland within a study extent influences species response to change, and the relationship between initial area and loss varies depending on replacement land-cover type

(Figure 8). Perhaps areas with more habitat present initially are used by greater diversity and abundance of species, and perhaps their response to disturbance is more extreme due to greater competition for remaining suitable habitat. It is also possible that losing a similar proportion of area from a small wetland could be more detrimental than loss from large wetlands for certain species, due to reduction of small wetlands to sizes that cannot reliably support breeding pairs. Although many studies on the importance of wetland size and wetland loss exist, I have found little literature exploring the relationship between the two (Mitsch and Gosselink, 2000).

I found that loss of wetland to developed land was the most important type of land-cover change included in my analysis. Among the tops models for each species, it appeared more frequently than any other type of wetland loss or overall land cover change. It partially explained the variation in change in mean abundance for nine of the 10 species, and related negatively to difference in abundance for seven species. Furthermore, it explained far more variation than loss of wetland to agriculture despite agriculture accounting for significantly more overall land area and wetland loss on average at both extents. The detrimental effects of urban development on wetlands are well-documented (Eppink et al., 2004; Faulkner, 2005).

The standard deviations of the random effect for ecoregion never exceeded 0.86 for any species. Therefore, it is likely that species responded in the same way to the fixed effects in their models across their range. It does not appear that the difference in mean abundance was heavily affected by the primary ecosystem around the routes.

My study was limited by the impact of detection probability on the BBS data. Point count surveys are subject to bias due to variation in detection probability. The amount of birds counted represent an unknown proportion of the total birds present. This detection probability relies heavily on the ability of observers to count and identify individuals effectively, which influenced by many factors. Variation in skill, experience, and hearing and sight ability among observers can affect detection, as well as the weather conditions under which a route is surveyed (Buckland et al., 2004). Although I did my best to remove sources of detection bias in the data cleaning process, it would be prudent to develop a method to adjust the count data by detection probability in future studies on this topic.

The power of my study was also limited by the relatively small number of routes I was able to include. BBS routes are sampled inconsistently due to low numbers of observers, so the quantity of data was limited to start with, and decreased further when I removed observations taken under suboptimal conditions. This low abundance of suitable data was one of the reasons I drew observations for each period from such a wide range of years. This practice could have introduced additional variation in the mean bird abundance within a time period.

**Conclusion**

My findings have a couple of implications for conservation of avian populations in prairie wetlands. First, not all types of habitat loss are created equal. Loss of wetland to development was more important to explaining the difference in mean abundance for more species than any other type of LULC change or wetland transition. Some species even responded positively to some types of wetland loss, depending on the type of land cover replacing the wetland and the amount of wetland available in the area to start with. Wetland loss should be avoided in general, but conservation strategies may be more effective if some types of wetland loss are more heavily restricted. For example, increasing urban development adjacent to and infringing on prairie wetlands should be limited more severely than agriculture. Furthermore, management strategies should be adjusted based on the size of wetlands or the total amount of wetland in a given area, because bird populations in areas with large amounts of wetland area likely respond differently to loss of habitat or nearby LULC change than those in small wetlands.

My study raises questions that could help to further our understanding of prairie wetland birds and how they interact with their landscape. For example, it might be useful to separate initial amounts of wetland in a LULC matrix into a few categories, to allow management plans to be adjusted based on the size or amount of wetland available for species to use. It would also be interesting to study wetland bird guilds in a study similar to mine, to determine whether members of the same guild exhibit similar responses to LULC change and wetland loss. As human-caused LULC changes such as increased urban sprawl, wetland inundation, and agricultural expansion continue to grow throughout the United States, understanding how species respond to different types of habitat change and loss will only become more important to consider for their conservation and management.

## References


Anderson, J. R., Hardy, E. E., Roach, J. T. and Witmer, R. E. (1976). A Land Use and Land Cover Classification System for Use with Remote Sensor Data. *U.S. Geological Survey* Professional Paper 964(28).

Bates, D., Maechler, M., Bolker, B., and Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, 67(1), 1-48. doi:10.18637/jss.v067.i01

Brown, M., and Dinsmore, J. (1986). Implications of Marsh Size and Isolation for Marsh Bird Management. *The Journal of Wildlife Management, 50*(3), 392-397. doi:10.2307/3801093

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., J.R. Poulsen, Stevens, M.H.H., and White, J.S.S. (2009). Generalized linear mixed models: a practical guide for ecology and evolution. *Trends Ecol. Evol.*, 24, 127-135. doi:10.1016/j.tree.2008.10.008

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., and Thomas, L.P. (2004). Advanced distance sampling. Oxford University Press, Oxford, UK.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach, 2nd ed. Sprinter Verlag New York, USA.

Chamberlain, D. , Fuller, R. , Bunce, R. , Duckworth, J. and Shrubb, M. (2000). Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *Journal of Applied Ecology*, 37, 771-788. doi:10.1046/j.1365-2664.2000.00548.x

Coppedge, B.R., Engle, D.M., Masters, R.E., and Gregory, M.S. (2001). Avian response to landscape change in fragmented southern Great Plains grasslands. *Ecological Applications*, 11 , 47-59.

Drummond, M.A., Auch, R.F., Karstensen, K.A., Sayler, K.L., Taylor, J.L., and Loveland, T.R. (2012). Land change variability and human–environment dynamics in the United States Great Plains. *Land Use Policy,* 29(3), 710-723.

Dunning, J., Danielson, B., and Pulliam, H. (1992). Ecological Processes That Affect Populations in Complex Landscapes. *Oikos, 65*(1), 169-175. doi:10.2307/3544901

Engle, D., Criner, T., Boren, J., Masters, R., and Gregory, M. (1999). RESPONSE OF BREEDING BIRDS IN THE GREAT PLAINS TO LOW DENSITY URBAN SPRAWL. *Great Plains Research, 9*(1), 55-73. Retrieved from http://www.jstor.org/stable/23777881

Eppink, F.V., van den Bergh, J.C.J.M., and Rietveld, P. (2004). Modeling biodiversity and land use: urban growth, agriculture and nature in a wetland area. *Ecological Economics*, 51, 201-216.

ESRI 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

Faulkner, S. (2005). Urbanization impacts on the structure and function of forested wetlands. *Urban Ecosystems*, 7, 89–106.

Forcey, G. M., Thogmartin, W. E., Linz, G. M., Bleier, W. J. and McKann, P. C. (2011). Land use and climate influences on waterbirds in the Prairie Potholes. *Journal of Biogeography*, 38. 1694-1707. doi:10.1111/j.1365-2699.2011.02510.x

Fry, J.A., Coan, M.J., Homer, C.G., Meyer, D.K., and Wickham, J.D. (2009). Completion of the National Land Cover Database (NLCD) 1992–2001 Land Cover Change Retrofit product. U.S. Geological Survey Open-File Report 2008–1379.

Horn, J. H., M. L. Phillips, R. R. Koford, W. R. Clark, M. A. Sovada, and R. J. Greenwood. (2005). Landscape composition, patch size, and distance to edges: interactions affecting duck reproductive success. *Ecological Applications,* 15, 1367–1376.

Jin, S., Yang, L., Danielson, P., Homer, C., Fry, J., and Xian, G. (2013). A comprehensive change detection method for updating the National Land Cover Database to circa 2011. *Remote Sensing of Environment*, 132, 159 – 175.

Johnson, N. (1949). Systems of Frequency Curves Generated by Methods of Translation. *Biometrika, 36*(1/2), 149-176. doi:10.2307/2332539

Mazerolle, M.J. (2019). AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). R package version 2.2-1. https://cran.r-project.org/package=AICcmodavg.

Matthews, S. N., Iverson, L. R., Prasad, A. M. and Peters, M. P. (2011). Changes in potential habitat of 147 North American breeding bird species in response to redistribution of trees and climate following predicted climate change. *Ecography*, 34, 933-945. doi:10.1111/j.1600-0587.2011.06803.x

Mitsch, W.J., Gosselink, J.G. (2000). The value of wetlands: importance of scale and landscape setting. *Ecological Economics.* 35, 25–33.

Mogollón, B., Frimpong, E.A., Hoegh, A.B., and Angermeier, P.L. (2016). Recent Changes in Stream Flashiness and Flooding, and Effects of Flood Management in North Carolina and Virginia. *Journal of the American Water Resources Association (JAWRA)*, 52( 3), 561– 577. doi: 10.1111/1752□1688.12408

Murkin, H.R., Kaminski, R.M., and Titman, R.D. (1982). Responses by dabbling ducks and aquatic invertebrates to an experimentally manipulated cattail marsh. *Canadian Journal of Zoology,* 60, 2324–2332.

Pardieck, K. (2001). Instructions for conducting the North American Breeding Bird Survey. U.S. Geological Survey Patuxent Wildlife Research Center, Laurel , Maryland.

Pardieck, K.L., D.J. Ziolkowski Jr., M. Lutmerding and M.-A.R. Hudson. (2018). North American Breeding Bird Survey Dataset 1966 - 2017, version 2017.0. U.S. Geological Survey, Patuxent Wildlife Research Center. https://doi.org/10.5066/F76972V8.

Petersen, K.L. and A.S Westmark. (2013). Bird use of wetlands in a midwestern metropolitan area in relation to adjacent land cover. *American Midland Naturalist*, 169(1), 221–228.

Price, C.V., Nakagaki, N., Hitt, K.J., and Clawges, R.M. (2003). Mining GIRAS—improving on a national treasure of land use data. In *Proceedings of the 23rd ESRI International Users Conference, July 7–11*, 2003, Redlands, California.

Price, S.J., Marks, D.R., Howe, R.W., Hanowski, J.M., and Niemi, G.J. (2004). The importance of spatial scale for conservation and assessment of anuran populations in coastal wetlands of the western Great Lakes, USA. *Landscape Ecology*, 20, 441-454.

R Core Team. (2013). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Rodewald, P. (Editor). (2015). The Birds of North America: https://birdsna.org. Cornell Laboratory of Ornithology, Ithaca, NY.

Rooney, R.C., Bayley, S.E., Creed, I.F., and Wilson, M.J. (2012). The accuracy of land cover-based wetland assessments is influenced by landscape extent. *Landscape Ecology*, 27, 1321–1335.

Sauer, J. R., D. K. Niven, J. E. Hines, K. L. Pardieck, J. E. Fallon, W. A. Link, and D. J. Ziolkowski, Jr. (2017). The North American Breeding Bird Survey, Results and Analysis 1966 - 2015. Version 12.23.2015 USGS Patuxent Wildlife Research Center, Laurel, MD

Sauer, J., Peterjohn, B., and Link, W. (1994). Observer Differences in the North American Breeding Bird Survey. *The Auk,* 111(1), 50-62. doi:10.2307/4088504

Scholtz, R., Polo, J., Fuhlendorf, S., Duckworth, G. (2017). Land cover dynamics influence distribution of breeding birds in the Great Plains, USA. *Biological Conservation*, 209, 323-331.

Semlitsch, R. D. and Bodie, J. R. (1998). Are Small, Isolated Wetlands Expendable?. *Conservation Biology*, 12, 1129-1133. doi:10.1046/j.1523-1739.1998.98166.x

Shutler, D. , Mullie, A. and Clark, R. G. (2000), Bird Communities of Prairie Uplands and Wetlands in Relation to Farming Practices in Saskatchewan. Conservation Biology, 14: 1441-1451. doi:10.1046/j.1523-1739.2000.98246.x

Sleeter, B.M., Sohl, T.L., Loveland, T.R., Auch, R.F., Acevedo, W., Drummond, M.A., Sayler, K.L., and Stehman, S.V. (2013). Land-cover change in the conterminous United States from 1973 to 2000. *Global Environmental Change Part A*, 23, 733-748.

Slifker, J.F., and Shapiro, S.S. (1980). The Johnson System: Selection and Parameter Estimation. *Technometrics*, 22(2), 239-246. doi: 10.1080/00401706.1980.10486139

Schmiegelow, F.K.A., Mönkkönen, M. (2002). Habitat loss and fragmentation in dynamic landscapes: avian perspectives from the boreal forest. *Ecological Applications,* 12, 375–89.

T Findlay, C. S. and Bourdages, J. (2000). Response Time of Wetland Biodiversity to Road Construction on Adjacent Lands. *Conservation Biology*, 14, 86-94. doi:10.1046/j.1523-1739.2000.99086.x

Thompson, S.J., Johnson, D.H., Niemuth, N.D., and Ribic, C.A. (2015). Avoidance of unconventional oil wells and roads exacerbates habitat loss for grassland birds in the North American Great Plains. *Biological Conservation*, 192, 82-90. doi:10.1016/j.biocon.2015.08.040

Vitousek, P. M., Mooney, H. A., Lubchenco, J. and Melillo, J. M. (1997). Human domination of Earth's ecosystems. *Science* 277, 494–499.

U.S. Geological Survey, 20140331, NLCD 2011 Land Cover (2011 Edition): None None, U.S. Geological Survey, Sioux Falls, SD.

Wiens, J. A., Rotenberry, J. T. and Van Horne, B. (1986). A Lesson in the Limitations of Field Experiments: Shrubsteppe Birds and Habitat Alteration. *Ecology*, 67, 365-376. doi:10.2307/1938579

## Appendix 1

Table 1: Sora. Models in gray are the better extents for each land cover type.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 64.44 | 0 | 0.33 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 64.74 | 0.3 | 0.28 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 64.84 | 0.4 | 0.27 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 67.84 | 3.4 | 0.06 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 69.57 | 5.13 | 0.03 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 69.83 | 5.39 | 0.02 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 74.46 | 10.02 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 74.94 | 10.5 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 75.05 | 10.61 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 75.07 | 10.63 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC |
| 75.89 | 11.45 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 76.67 | 12.23 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 77.32 | 12.88 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 80.07 | 15.63 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 81.52 | 17.08 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 82.45 | 18.01 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 82.66 | 18.22 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 84.25 | 19.81 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 84.73 | 20.29 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 85 | 20.56 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 85.41 | 20.97 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 86.19 | 21.75 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 95.65 | 31.21 | 0 | null |

Table 2: Sedge Wren. Models in gray are the better extents for each land cover type.

| AICc | ΔAICc | AICcWt | |
|---|---|---|---|
| 67.28 | 0 | 0.43 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 68.7 | 1.42 | 0.21 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 69.28 | 2 | 0.16 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 69.99 | 2.72 | 0.11 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 71.4 | 4.12 | 0.06 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 74.13 | 6.85 | 0.01 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 75.2 | 7.92 | 0.01 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 77.27 | 10 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC |
| 79.16 | 11.88 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 80.15 | 12.87 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 81.42 | 14.14 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 82.89 | 15.61 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 83.07 | 15.79 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 83.94 | 16.66 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 85.54 | 18.26 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 86.65 | 19.38 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 86.85 | 19.57 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 87.02 | 19.74 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 87.55 | 20.27 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 87.96 | 20.68 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 88.23 | 20.96 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 88.8 | 21.52 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 100.33 | 33.05 | 0 | null |

Table 3: Yellow-headed Blackbird. Models in gray are the better extents for each land cover type.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 75.06 | 0 | 0.69 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 77.42 | 2.36 | 0.21 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 79.55 | 4.49 | 0.07 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 83 | 7.94 | 0.01 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 83.54 | 8.49 | 0.01 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 86.3 | 11.24 | 0 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 89.09 | 14.03 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 89.9 | 14.85 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 90.58 | 15.52 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 91.84 | 16.78 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 93.77 | 18.71 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 95.15 | 20.09 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 95.59 | 20.53 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 96.2 | 21.15 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 97.22 | 22.16 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 97.89 | 22.83 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 98.3 | 23.24 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 99.83 | 24.78 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 100.16 | 25.1 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 100.23 | 25.17 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 100.95 | 25.89 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 103.3 | 28.24 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 112.61 | 37.55 | 0 | null |

Table 4: Blue-winged Teal. Models in gray are the better extents for each land cover type.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 90.24 | 0 | 0.54 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 92.22 | 1.98 | 0.2 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 94.16 | 3.92 | 0.08 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 94.44 | 4.2 | 0.07 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 94.69 | 4.45 | 0.06 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 96.19 | 5.94 | 0.03 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 97.58 | 7.34 | 0.01 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 98.77 | 8.53 | 0.01 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 99.94 | 9.7 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 100.11 | 9.87 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 100.82 | 10.58 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 101.4 | 11.16 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 103.4 | 13.16 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 103.55 | 13.31 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 105.53 | 15.29 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 105.85 | 15.61 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 106.41 | 16.16 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 108.16 | 17.91 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 108.65 | 18.4 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 109.33 | 19.09 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 109.46 | 19.21 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 111.54 | 21.29 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 119.61 | 29.36 | 0 | null |

Table 5: Wood Duck. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 139.48 | 0 | 0.38 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 140.11 | 0.63 | 0.28 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 140.36 | 0.88 | 0.25 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 143.57 | 4.09 | 0.05 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 145.09 | 5.61 | 0.02 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 146.9 | 7.42 | 0.01 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 148.3 | 8.82 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 149.05 | 9.57 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 150.13 | 10.65 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 150.21 | 10.74 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 151.02 | 11.54 | 0 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 151.6 | 12.12 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 151.82 | 12.35 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 153.27 | 13.79 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 154.64 | 15.16 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 155.97 | 16.49 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 156.84 | 17.36 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 157.19 | 17.71 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 159.64 | 20.16 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 161.37 | 21.89 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 163.9 | 24.43 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 164.54 | 25.07 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 172.01 | 32.53 | 0 | null |

Table 6: Canada Goose. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 195.57 | 0 | 0.73 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 197.98 | 2.4 | 0.22 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 203.05 | 7.47 | 0.02 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 204.06 | 8.49 | 0.01 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 204.78 | 9.21 | 0.01 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 205.48 | 9.91 | 0.01 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 205.92 | 10.34 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 206.37 | 10.8 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 206.49 | 10.91 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 208 | 12.42 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 209.12 | 13.55 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 210.97 | 15.4 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 210.99 | 15.42 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 211.7 | 16.13 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 212.27 | 16.69 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 215.35 | 19.78 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 215.73 | 20.15 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 218.25 | 22.68 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 222.31 | 26.74 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 222.86 | 27.29 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 222.9 | 27.33 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 224.15 | 28.57 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 232.9 | 37.32 | 0 | null |

Table 7: Common Yellowthroat. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 243.25 | 0 | 0.56 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 246.31 | 3.06 | 0.12 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 246.37 | 3.11 | 0.12 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 248.09 | 4.83 | 0.05 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 248.29 | 5.04 | 0.05 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 248.66 | 5.41 | 0.04 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 248.73 | 5.48 | 0.04 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 251.27 | 8.02 | 0.01 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 251.74 | 8.48 | 0.01 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 253.4 | 10.15 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 254.4 | 11.15 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 254.49 | 11.24 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 255.3 | 12.05 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 258.77 | 15.52 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 259.21 | 15.95 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 259.23 | 15.98 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 261.88 | 18.62 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 262.11 | 18.86 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 262.87 | 19.62 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 264.97 | 21.71 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 265.19 | 21.93 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 267.23 | 23.98 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 275.03 | 31.78 | 0 | null |

Table 8: Mallard. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 256 | 0 | 0.76 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 261.04 | 5.04 | 0.06 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 261.34 | 5.34 | 0.05 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 261.39 | 5.39 | 0.05 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 263.12 | 7.12 | 0.02 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 263.72 | 7.71 | 0.02 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 263.74 | 7.74 | 0.02 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 264.82 | 8.82 | 0.01 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 265.48 | 9.47 | 0.01 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 269.06 | 13.06 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 269.68 | 13.68 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 270.36 | 14.35 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 271.92 | 15.92 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 277.26 | 21.26 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 277.32 | 21.31 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 277.89 | 21.88 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 279.39 | 23.39 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 280.32 | 24.31 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 280.62 | 24.61 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 282.38 | 26.38 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 283.26 | 27.26 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 284.66 | 28.66 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 293.6 | 37.6 | 0 | null |

Table 9: Common Grackle. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 284.82 | 0 | 0.85 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 290.67 | 5.85 | 0.05 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 290.92 | 6.1 | 0.04 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 291.76 | 6.94 | 0.03 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 292.37 | 7.55 | 0.02 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 292.56 | 7.74 | 0.02 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 297.27 | 12.45 | 0 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 297.33 | 12.51 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 297.73 | 12.91 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 298.54 | 13.72 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 300.16 | 15.35 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 301.23 | 16.41 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 303.49 | 18.67 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 304.22 | 19.4 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 304.5 | 19.69 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 305.28 | 20.46 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 306.5 | 21.68 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 307.1 | 22.29 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 309.95 | 25.14 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 310.96 | 26.15 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 311.39 | 26.57 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 311.63 | 26.81 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 317.97 | 33.16 | 0 | null |

Table 10: Red-winged Blackbird. Models in gray are the better extents for each land cover type. These models were carried forward for inclusion in more complex models.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| 318.37 | 0 | 0.53 | Init. WetlandB+ LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) |
| 320.37 | 1.99 | 0.2 | Init. WetlandB+ LosstoWaterB + (Init. Wetland*LosstoWaterB) |
| 320.87 | 2.5 | 0.15 | Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) |
| 323.03 | 4.66 | 0.05 | Init. WetlandB+ ΔWetlandB + (Init. Wetland*ΔWetlandB) |
| 323.56 | 5.19 | 0.04 | Init. WetlandB+ LosstoGrassB + (Init. Wetland*LosstoGrassB) |
| 325.62 | 7.25 | 0.01 | Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) |
| 326.39 | 8.02 | 0.01 | Init. WetlandB+ LosstoForestB + (Init. Wetland*LosstoForestB) |
| 329.19 | 10.82 | 0 | Init. WetlandB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) |
| 331.11 | 12.74 | 0 | Init. WetlandHUC + LosstoGrassHUC + (Init. WetlandHUC*LosstoGrassHUC) |
| 331.56 | 13.19 | 0 | Init. WetlandHUC + LosstoForestHUC + (Init. WetlandHUC*LosstoForestHUC) |
| 332.95 | 14.58 | 0 | Init. WetlandHUC + ΔWetlandHUC + (Init. WetlandHUC*ΔWetlandHUC) |
| 334.04 | 15.67 | 0 | Init. WetlandHUC + LosstoWaterHUC + (Init. WetlandHUC*LosstoWaterHUC) |
| 337.04 | 18.67 | 0 | Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) |
| 337.8 | 19.43 | 0 | Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) |
| 338.01 | 19.64 | 0 | Init. WaterHUC + ΔWaterHUC + (Init. WaterHUC*ΔWaterHUC) |
| 338.35 | 19.98 | 0 | Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) |
| 340.05 | 21.68 | 0 | Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) |
| 340.58 | 22.21 | 0 | Init. DevHUC + ΔDevelopedHUC + (Init. DevHUC*ΔDevelopedHUC) |
| 342.89 | 24.52 | 0 | Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) |
| 345.04 | 26.66 | 0 | Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) |
| 345.21 | 26.84 | 0 | Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) |
| 346.37 | 28 | 0 | Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) |
| 352.89 | 34.52 | 0 | null |

Table 11: Wood Duck, complex models. Models in bold contributed to averaged model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **122.37** | **0** | **0.55** | **Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB)** |
| **123.91** | **1.54** | **0.25** | **Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB)** |
| **125.7** | **3.33** | **0.1** | **Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB)** |
| **126.1** | **3.73** | **0.09** | **Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB)** |
| 132.29 | 9.92 | 0 | Init. WetlandB+ ΔWetlandB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 133.32 | 10.95 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 133.68 | 11.31 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) + (Init. Wetland*ΔWetlandB) |
| 138.42 | 16.05 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 143.12 | 20.75 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) |
| 146.26 | 23.89 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 172.01 | 49.64 | 0 | ull |

Table 12: Canada Goose, complex models. Model in bold reported as strongest model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **166** | **0** | **0.99** | **Init. WetlandB+ ΔWetlandB + Init. WetlandHUC + LosstoDevelopedHUC + (Init. WetlandHUC*LosstoDevelopedHUC) + (Init. Wetland*ΔWetlandB)** |
| 178.53 | 12.52 | 0 | Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) |
| 178.71 | 12.71 | 0 | Init. WetlandB+ ΔWetlandB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 179.45 | 13.45 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 181.14 | 15.14 | 0 | Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) |
| 181.24 | 15.23 | 0 | Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) |
| 183.22 | 17.22 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 186.14 | 20.13 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) + (Init. Wetland*ΔWetlandB) |
| 193.77 | 27.77 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 193.85 | 27.84 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) |
| 232.9 | 66.89 | 0 | null |

Table 13: Common Yellowthroat, complex models. Models in bold contributed to averaged model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **197.98** | **0** | **0.62** | **Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| **200.38** | **2.39** | **0.19** | **Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| **200.89** | **2.91** | **0.14** | **Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| 203.8 | 5.82 | 0.03 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 205.45 | 7.47 | 0.01 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 209.41 | 11.43 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 213.04 | 15.05 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 217.37 | 19.39 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 220.9 | 22.92 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 222.78 | 24.79 | 0 | Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 225.56 | 27.58 | 0 | Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) |
| 226.05 | 28.07 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) + (Init. WetlandHUC*ΔWetlandHUC) |
| 228.5 | 30.52 | 0 | Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) |
| 228.92 | 30.94 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 229.85 | 31.87 | 0 | Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) |
| 233.38 | 35.39 | 0 | Init. WetlandB+ ΔWetlandB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 237.37 | 39.38 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 242.18 | 44.19 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) + (Init. Wetland*ΔWetlandB) |
| 244.83 | 46.85 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 275.03 | 77.05 | 0 | null |

Table 14: Mallard, complex models. Models in bold contributed to averaged model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **199.33** | **0** | **0.68** | **Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB)+ (Init. WaterB*ΔWaterB)** |
| **202.25** | **2.93** | **0.16** | **Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| **202.65** | **3.32** | **0.13** | **Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| 206.08 | 6.76 | 0.02 | Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 208.55 | 9.23 | 0.01 | Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 214.13 | 14.81 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 214.4 | 15.07 | 0 | Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) + (Init. WetlandHUC*ΔWetlandHUC)+ (Init. WaterB*ΔWaterB) |
| 220.34 | 21.02 | 0 | Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) + (Init. Wetland*ΔWetlandB)+ (Init. WaterB*ΔWaterB) |
| 221.14 | 21.82 | 0 | Init. WaterB + ΔWaterB + Init. WetlandB+ ΔWetlandB + Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) + (Init. Wetland*ΔWetlandB)+ (Init. WaterB*ΔWaterB) |
| 225.82 | 26.5 | 0 | Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 226.37 | 27.05 | 0 | Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) |
| 228.3 | 28.97 | 0 | Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) |
| 229.31 | 29.99 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 232.61 | 33.28 | 0 | Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) |
| 235.35 | 36.02 | 0 | Init. WetlandB+ ΔWetlandB + Init. WetlandHUC + LosstoAgricultureHUC + (Init. WetlandHUC*LosstoAgricultureHUC) + (Init. Wetland*ΔWetlandB) |
| 240.63 | 41.31 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) + (Init. WetlandHUC*ΔWetlandHUC) |
| 240.64 | 41.31 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 245.61 | 46.29 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) + (Init. Wetland*ΔWetlandB) |
| 247.25 | 47.93 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassHUC + ΔGrassHUC + (Init. GrassHUC*ΔGrassHUC) + (Init. Wetland*ΔWetlandB) |
| 293.6 | 94.27 | 0 | null |

Table 15: Common Grackle, complex models. Model in bold reported as strongest model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **246.02** | **5.32** | **0.06** | **Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| 248.5 | 7.8 | 0.02 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 248.73 | 8.02 | 0.02 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 254.18 | 13.47 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 254.52 | 13.82 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) + (Init. WetlandHUC*ΔWetlandHUC) + (Init. WaterB*ΔWaterB) |
| 259.05 | 18.34 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 264.74 | 24.03 | 0 | Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 264.99 | 24.29 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 266.29 | 25.59 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 270.75 | 30.04 | 0 | Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) |
| 272.5 | 31.8 | 0 | Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) |
| 272.7 | 32 | 0 | Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) |
| 273.24 | 32.53 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 277.64 | 36.94 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestHUC + ΔForestHUC + (Init. ForestHUC*ΔForestHUC) + (Init. Wetland*ΔWetlandB) |
| 277.66 | 36.95 | 0 | Init. WetlandB+ ΔWetlandB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 284.01 | 43.31 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 290.77 | 50.07 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 291.15 | 50.44 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) |
| 317.97 | 71.95 | 0 | null |

Table 16: Red-winged Blackbird, complex models. Model in bold reported as strongest model.

| AICc | ΔAICc | AICcWt | Parameters |
|---|---|---|---|
| **256.51** | **0** | **0.94** | **Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB)** |
| 262.41 | 5.9 | 0.05 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 266.19 | 9.68 | 0.01 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 269 | 12.49 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 269.33 | 12.81 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 276.26 | 19.75 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ Init. AgricultureB + ΔAgricultureB + (Init. AgricultureB*ΔAgricultureB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 280.89 | 24.38 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 286.15 | 29.63 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 286.42 | 29.91 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB+ Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUC*ΔAgricultureHUC) + (Init. Wetland*ΔWetlandB) + (Init. WaterB*ΔWaterB) |
| 287.05 | 30.54 | 0 | Init. WetlandB+ ΔWetlandB + LosstoDevelopedB + (Init. Wetland*LosstoDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 289.43 | 32.92 | 0 | Init. WetlandB+ ΔWetlandB + LosstoWaterB + (Init. Wetland*LosstoWaterB) + (Init. Wetland*ΔWetlandB) |
| 292.75 | 36.24 | 0 | Init. WetlandB+ ΔWetlandB + Init. WaterB + ΔWaterB + (Init. WaterB*ΔWaterB) + (Init. Wetland*ΔWetlandB) |
| 296.73 | 40.22 | 0 | Init. WetlandB+ ΔWetlandB + LosstoGrassB + (Init. Wetland*LosstoGrassB) + (Init. Wetland*ΔWetlandB) |
| 298.74 | 42.22 | 0 | Init. WetlandB+ ΔWetlandB + LosstoForestB + (Init. Wetland*LosstoForestB) + (Init. Wetland*ΔWetlandB) |
| 299.47 | 42.96 | 0 | Init. WetlandB+ ΔWetlandB + LosstoAgricultureB + (Init. Wetland*LosstoAgricultureB) + (Init. Wetland*ΔWetlandB) |
| 306.07 | 49.56 | 0 | Init. WetlandB+ ΔWetlandB + Init. ForestB + ΔForestB + (Init. ForestB*ΔForestB) + (Init. Wetland*ΔWetlandB) |
| 310.68 | 54.17 | 0 | Init. WetlandB+ ΔWetlandB + Init. DevelopedB + ΔDevelopedB + (Init. DevelopedB*ΔDevelopedB) + (Init. Wetland*ΔWetlandB) |
| 315.2 | 58.69 | 0 | Init. WetlandB+ ΔWetlandB + Init. AgricultureHUC + ΔAgricultureHUC + (Init. AgricultureHUCΔ*ΔAgricultureHUC) + (Init. Wetland*ΔWetlandB) |
| 315.7 | 59.18 | 0 | Init. WetlandB+ ΔWetlandB + Init. GrassB + ΔGrassB + (Init. GrassB*ΔGrassB) + (Init. Wetland*ΔWetlandB) |
| 352.89 | 96.38 | 0 | null |